

1-1-2009

## Developing vegetation metrics for the assessment of beneficial uses of impounded wetlands surrounding Great Salt Lake, Utah, USA

Heidi M. Hoven

*Institute for Watershed Sciences, Kamas, UT*

Theron G. Miller

*Utah Division of Water Quality, Salt Lake City*

Follow this and additional works at: <http://digitalcommons.usu.edu/nrei>

### Recommended Citation

Hoven, Heidi M. and Miller, Theron G. (2009) "Developing vegetation metrics for the assessment of beneficial uses of impounded wetlands surrounding Great Salt Lake, Utah, USA," *Natural Resources and Environmental Issues*: Vol. 15, Article 11.

Available at: <http://digitalcommons.usu.edu/nrei/vol15/iss1/11>

This Article is brought to you for free and open access by the Quinney Natural Resources Research Library, S.J. and Jessie E. at DigitalCommons@USU. It has been accepted for inclusion in Natural Resources and Environmental Issues by an authorized administrator of DigitalCommons@USU. For more information, please contact [becky.thoms@usu.edu](mailto:becky.thoms@usu.edu).



# Developing Vegetation Metrics for the Assessment of Beneficial Uses of Impounded Wetlands Surrounding Great Salt Lake, Utah, USA

Heidi M. Hoven<sup>1</sup> & Theron G. Miller<sup>2</sup>

<sup>1</sup> The Institute for Watershed Sciences, 1937 Mirror Lake Hwy, Kamas, UT 84036 USA, <sup>2</sup> Utah Department of Environmental Quality, Division of Water Quality, 288 North 1460 West, PO Box 144870, Salt Lake City, UT 84114 USA

Corresponding author:

Heidi M. Hoven, The Institute for Watershed Sciences, 1937 Mirror Lake Hwy, Kamas, UT 84036, USA

E-mail: hmhoven@iwsinciences.org

## ABSTRACT

Many wetlands around Farmington Bay of Great Salt Lake are managed waterfowl habitat by means of impounding the flow at the terminus of the Jordan River. The majority of the Jordan River flow is comprised of the secondary-treated effluent of several municipal waste water treatment plants (WWTP), resulting in elevated phosphorus concentrations. This study was initiated to determine whether the assimilative capacity for phosphorus of the impounded wetlands has been exceeded, resulting in a negative impact to the submerged aquatic vegetation (SAV) in the wetlands. The majority of the SAV is sago pondweed and western fineleaf pondweed (*Stuckenia pectinata* and *S. filiformis* ssp. *occidentalis*, respectively), highly preferred food items for waterfowl. Utah Department of Environmental Quality has identified support for waterfowl and shorebirds and the aquatic life in their food chain as the primary beneficial use of these wetlands, and thus, loss or degradation of *Stuckenia* prior to waterfowl fall staging and migration may constitute a loss of this important beneficial use. Therefore, Utah needs vegetation metrics that will indicate relative health of a wetland with respect to the abundance, density and health of the SAV and the level of nutrient loading it receives. The primary goal of this ongoing study is to develop wetland assessment methods that will be used to establish water quality standards and methods for Clean Water Act §305(b)/303(d) assessments—one of the first attempts by any state of the U.S. to set wetland water quality standards through development of site-specific assessment protocols. To develop metrics that describe the relationship between nutrient gradients and biological responses, we are 1) testing potentially useful parameters for their utility in assessing wetland condition; and 2) refining condition metrics that will identify thresholds of significant change (impairment) that can be attributed to nutrients. This paper presents the first of several potentially useful vegetation metrics. Our analyses showed that percent areal cover of SAV in nutrient enriched wetlands senesced 62-84% from July through November whereas the vegetation in a non-impacted reference wetland remained stable. The fall senescence occurs at a time when migratory waterfowl rely on submerged aquatic vegetation (SAV) for sustenance.



**Figure 1**—Eastern shore of Great Salt Lake, USA showing all impounded wetland sites of the State of Utah Division of Water Quality’s study on ecological and beneficial use assessment of Farmington Bay wetlands. Reference sites are located at the PSG (Public Shooting Grounds) and nutrient enriched sites are located at FB WMA (Farmington Bay Wildlife Management Area), NEW (Newstate Duck Club), AMB (Ambassador Duck Club), and ISSR (Inland Sea Shorebird Reserve).

## INTRODUCTION

Great Salt Lake, Utah, USA (located between 40° and 41° N, 113° and 112° W) is the fourth largest terminal lake in the world (Figure 1). On average, the lake is 3 to 5 times the salinity of the ocean. Yet, the very gentle slopes at the lake’s margin provide for approximately  $1.82 \times 10^5$  ha ( $4.5 \times 10^5$  acres) of fresh to slightly brackish lacustrine wetlands leading from tributary flows. These vast acreages of wetlands, occurring in the interior, arid US west, are vitally important to the five to seven million waterfowl and shorebirds that depend on Great Salt Lake and its wetlands for nesting and migratory staging each year. About  $6.1 \times 10^4$  ha of wetlands occur in Farmington Bay (situated

on the southeastern portion of the lake), of which approximately  $3.5 \times 10^4$  ha are impounded and managed for waterfowl. All of these impounded wetlands are supplied by the Jordan River, the majority flow of which is provided by the effluent of four major municipal waste water treatment plants. The resulting ambient P concentration ranges from 0.9 to 1.3 mg l<sup>-1</sup> (Miller & Hoven 2007). Nearly all of the Jordan River water flows through impounded wetlands before being released to the open water of Farmington Bay. It is not known how the high nutrient levels in these wetlands impact this ecosystem.

The Federal Clean Water Act requires all states to assign “beneficial uses” to all surface waters of the US and to perform frequent assessments of the water quality to determine if these beneficial uses are being met. As such, the Utah Department of Environmental Quality assigned “support for waterfowl and shorebirds and the aquatic life in their food chain for Great Salt Lake wetlands” as a beneficial use of these wetlands. The areas of concern for this study are the nutrient enriched impoundments supported by the Jordan River (Figure 1). Wetlands located at the north end of the lake (Public Shooting Grounds) where nutrient enrichment is far less serve as reference sites (Figure 1). Both nutrient enriched and reference impoundments are managed nearly exclusively for the production of submerged aquatic vegetation (SAV) and in particular, for sago pondweed (*Stuckenia* sp.), because it is the preferred forage taxon by omnivorous waterfowl (Kantrud 1990). Although the impoundments are managed to optimize SAV growth for waterfowl, the wetlands are also important for other waterbirds and wildlife (e.g., Olson et al. 2004). One wetland at the southern end of Farmington Bay (The Inland Sea Shorebird Reserve (ISSR)) is specifically managed for shorebirds but its impoundments are deep enough to grow SAV and attract waterfowl (Hoven & Miller, personal observation).

SAV has been shown to be a sensitive indicator of water quality (Kemp et al. 1983; Orthe & Moore 1983; Tomasko et al. 1996; Stumpf et al. 1999) and a sentinel accumulator (Wolfe et al. 1976; Burrell & Schubel 1977; Brix & Lyngby 1983; Ward 1987; Hoven et al. 1999) of anthropogenic stressors in shallow estuarine embayments worldwide. SAV provides a myriad of ecological functions to a watershed. It provides a protective environment and nursery function to invertebrates, fish, and shellfish; stabilizes sediments; cycles nutrients and elements; attenuates nutrients and other pollutants; and filters suspended sediments (Thayer et al. 1975, 1984; Kenworthy et al. 1982; Phillips & Meñez

1988). Because SAV have specific light requirements, they may be susceptible to shading by algae (epiphytes, macroalgal mats, and/or phytoplankton), duckweed, suspended sediments, and/or water color (Buzzelli et al. 1999; Hall et al. 1999; Stumpf et al. 1999; Zieman et al. 1999). Algal blooms are stimulated by increased nutrient loads and often associated with inputs from high human density and/or industrial areas or areas of agricultural runoff (Staver et al. 1996; Madden & Kemp 1996) and have been shown to correlate with decline in areal cover of seagrasses (Short & Burdick 1996; Valiela et al. 1997).

In the impounded wetlands of Farmington Bay we have observed several indicators of hypertrophy, including floating mats of filamentous green algae (primarily *Cladophora* spp.), cyanobacteria (primarily *Oscillatoria* spp.) and duckweed (*Lemna minor*) (Rushforth & Rushforth 2007). Biofilm was frequently observed at nutrient enriched sites on *Stuckenia* leaves, which was comprised of epiphytes (mostly diatoms), mucilaginous material and fine inorganic sediments (Rushforth & Rushforth 2007). Our study has been designed to 1) identify thresholds of adverse biological or ecological changes to gradients in nutrients and other parameters that are typically associated with hypertrophy, and 2) identify sensitive and ecologically important responses to nutrient enrichment in Farmington Bay and its wetlands. An array of these metrics could then be incorporated into an index of biological integrity that quantifies various ecological functions against a gradient in nutrients. Ultimately, thresholds along this scoring range will be used to establish beneficial use support status. This effort represents one of the first attempts by any state to establish water quality standards and methods for Federal Clean Water Act §305(b)/303(d) assessments for wetlands.

## METHODS

Five impounded sites were identified around or near Farmington and Bear River Bays of Great Salt Lake, Utah (Figure 1). Ambassador Duck Club, Newstate Duck Club, and Farmington Bay Wildlife Management Area (FB WMA) all receive water from the Jordan River and empty into a downstream duck club (Newstate Duck Club passes much of its water on to FB WMA) or releases it directly to Farmington Bay. The ISSR receives water from the Northpoint Consolidated Canal (a diversion from Jordan River water). The ISSR generally allows its impoundments to draw down (recede) via evaporation during summer months to provide forage and nesting habitat for shorebirds. Public Shooting Grounds (PSG) is situated at the north end

of the lake on Bear River Bay and receives its water from freshwater springs and some irrigation return flow. The ponds typically have a management protocol that calls for 46 cm (18 inches) of water depth to maximize SAV growth; however, during our study many ponds were frequently less than 46 cm, due to limited water availability.

Sample site selection was based on the assumption that nutrient assimilation would result in declining nutrient concentrations as nutrient enriched water flowed through successive ponds leading from the terminus of the Jordan River or from its water diversions serving the Ambassador Duck Club or the ISSR. In this manner we expected to describe co-located biological responses to a water column nutrient gradient. In reality, however, only one wetland complex, Ambassador Duck Club, exhibited a distinct nutrient gradient (Miller & Hoven 2007). Retention times in the Ambassador ponds were much greater than in the other pond systems, which most likely contributed to this uniqueness (Miller & Hoven 2007). Nevertheless, there was enough variation in nutrient concentrations among the various pond systems to address the nutrient enrichment and biological response relationship.

We measured nitrate-nitrite and total phosphorus using standard EPA methods (353.2 and 365.2, respectively). We also measured dissolved oxygen (DO), pH, temperature, and electrical conductivity (EC) using Hydrolab® or In-Situ® multiprobe sondes. Measurements and sample collections were performed at designated outlet culverts (easily identifiable landmarks) that were located near biological sample collections (indicated by site location points on Figure 1). The water quality sampling points were collected at approximately one month intervals during daylight hours from May through November, 2005. Sediment phosphorus was also determined for the upper 10 cm of sediment surface in all ponds using the persulfate digestion method EPA 365.1.

One-hundred-meter long transects were measured by pacing within each pond 50 m offshore and 100 m away from (when possible) culverts to avoid any influence of water flow patterns on the distribution of the plants. One square meter quadrats were established by laying two 2.0 m PVC poles 0.5 m apart and perpendicular to the transect line at 10 random locations along the transect. Percent cover (to the nearest 1%) as visual areal estimates at mid-canopy of the SAV was determined within each quadrat. The 2.0 m PVC poles were marked to show area designations

(e.g., 1, 5, 10, 25, 30%), a modification of the Daubenmire frame technique (Daubenmire 1959). Species composition and general observations of turbidity and algal or duckweed cover were also noted. It was not possible to use a Secchi disc for turbidity determinations due to the shallowness of the water. Percent cover was estimated during July, August, and November of 2005.

Three composite samples of the dominant species of SAV in each pond were collected for tissue carbon (as total organic carbon), nitrogen (as total nitrogen), and phosphorus (as total phosphorus) analysis. Plant samples were stored in a refrigerator in sealed plastic bags until they were processed. Processing included rinsing plants free of sediment and debris, wiping periphyton off with absorbent paper towels and hand selecting approximately 5 g (wet weight) bright green leaves with forceps. Leaves of similar length on a turion were used rather than shorter leaves on distal-most shoots in an attempt to collect similarly aged leaves. Leaves were kept under water while processing and once adequate sample was derived, the leaves were rinsed in distilled deionized water and dried in aluminum foil trays at 34°C for at least 72 hours. The dried samples were quickly placed in clean, labeled sealed plastic bags and stored in a closed box prior to chemical analysis. Total organic carbon and total nitrogen were determined using ASTM D5373 analytical methods at Timpview Analytical Laboratories of Orem, UT; and total phosphorus was determined using EPA Method 325.2 (ICP atomic emission spectroscopy) at the Brigham Young University Soils Lab. At the lab, samples from each site were composited to ensure adequate material for analysis.

Data were analyzed using multivariate factor analysis. Water quality factors were determined following the methods outlined by Madon (2006). Of the eight parameters used, TSS, conductivity, and temperature explained the least amount of variability when ordinated in the second and third factors and hence, were excluded to reduce the data to one ordination factor. All water quality data were transformed by Log10, using  $\text{Log}_{10}(x + 1)$  for zeros. Percent cover data were first composited as total SAV per quadrat (i.e., % *Stuckenia* sp. plus % *Ruppia cirrhosa* (spiral ditchgrass)) and transformed by arcsine $\sqrt{x}$ , using arcsine (square root  $((0 + 3/8) / (15 + 3/4))$ ) for zeros (Anscombe 1948). Univariate repeated measures were performed to assess whether the ponds were responding differently with respect to percent cover across time.

**Table 1**—Mean water quality parameters at the upper ponds of nutrient enriched and reference impoundments during 2005 (PSG are reference ponds). Values of 0.01 for total P or NO<sub>2</sub>-NO<sub>3</sub> are reported as 1/2 of the instrument detection limit of 0.02. DO = dissolved oxygen. TDS = total dissolved solids. se = standard error. nd = do data.

SPRING	DO		NO <sub>2</sub> + NO <sub>3</sub>		P		pH	se	TDS	
	(mg l <sup>-1</sup> )	se	(mg l <sup>-1</sup> )	se	(mg l <sup>-1</sup> )	se			(mg l <sup>-1</sup> )	se
AMB	11.1	1.10	0.66	0.2	0.50	0.05	9.0	0.1	666	n=1
FB WMA	8.6	0.10	0.02	0.0	0.20	0.05	9.5	0.4	1099	591
NEW	1.8	n=1	1.15	n=1	0.40	n=1	7.9	n=1	nd	
ISSR	16.7	0.03	0.01	0.0	0.10	0.05	10.1	0.2	1716	176
PSG	8.7	n=1	0.05	n=1	0.01	n=1	8.7	n=1	2702	n=1

SUMMER	DO		NO <sub>2</sub> + NO <sub>3</sub>		P		pH	se	TDS	
	(mg l <sup>-1</sup> )	se	(mg l <sup>-1</sup> )	se	(mg l <sup>-1</sup> )	se			(mg l <sup>-1</sup> )	se
AMB	9.1	1.7	1.70	0.3	0.80	0.030	9.1	0.4	nd	
FB WMA	6.2	0.1	0.02	0.0	0.10	0.010	9.3	0.2	861	126
NEW	1.6	0.2	2.00	0.9	0.70	0.300	8.1	0.2	nd	
ISSR	4.2	1.2	0.01	0.0	0.20	0.100	9.1	0.1	1113	208
PSG	9.0	2.3	0.01	0.0	0.04	0.005	9.0	0.1	1576	136

FALL	DO		NO <sub>2</sub> + NO <sub>3</sub>		P		pH	se	TDS	
	(mg l <sup>-1</sup> )	se	(mg l <sup>-1</sup> )	se	(mg l <sup>-1</sup> )	se			(mg l <sup>-1</sup> )	se
AMB	8.8	0.7	1.90	0.1	1.20	0.1	8.5	0.04	nd	
FB WMA	9.4	2.0	0.10	0.1	0.10	0.1	9.1	0.20	nd	
NEW	9.2	2.2	4.40	1.0	0.90	0.2	8.3	0.30	nd	
ISSR	9.0	1.6	0.01	0.0	0.30	0.1	9.1	0.10	1771	235
PSG	8.3	0.7	0.01	0.0	0.01	0.0	8.6	0.10	2896	275

**RESULTS**

*Stuckenia* sp. was the dominant SAV observed among the sites. *Ruppia cirrhosa* was occasionally present in the more-saline ponds and *Ceratophyllum demersum* (coon’s tail) was also occasionally present in small proportions but very rare. There were varying amounts of floating mats of filamentous green algae (primarily *Cladophora* spp.), the cyanobacterium *Oscillatoria* spp., among others, and epiphytic algae and diatoms were frequently on the SAV at the nutrient enriched sites (Rushforth & Rushforth 2007).

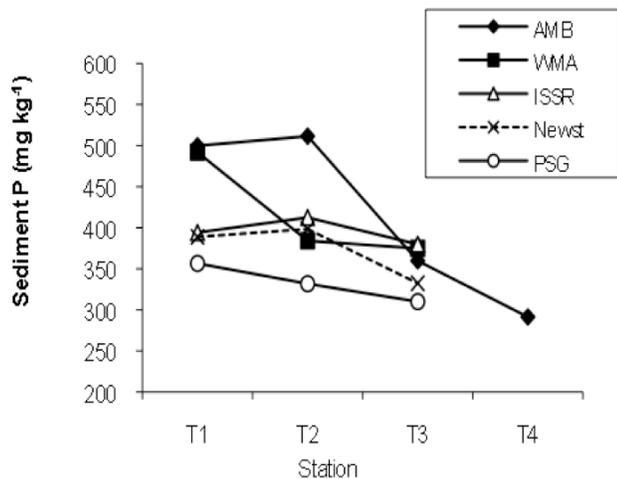
Among all ponds, water column pH occasionally exceeded Utah’s freshwater standard of 9.0 and this included reference ponds in PSG that had less than 0.02 mg l<sup>-1</sup> total P (Table 1). The daytime dissolved oxygen concentration in the PSG ponds was often 120% to 200% saturation. The source of high dissolved oxygen in this case, was likely the dense meadows of SAV and a calcareous green macroalga, *Chara* sp. Low nutrient and high dissolved oxygen concentrations with dense vegetative cover was a common condition among the impounded wetlands at our reference

site. Table 1 lists the means of water quality samples collected among the nutrient enriched and reference sites. Total P at the reference sites was often below the detection limit (0.02 mg l<sup>-1</sup>) while nutrient enriched wetland sites always contained substantially more P (0.10–0.90 mg l<sup>-1</sup>).

Mean sediment P concentrations measured in successive (upstream to downstream) ponds in the five different wetland complexes are summarized in Figure 2. There was at least a slight decrease in sediment concentrations in each of the systems, with the biggest declines in the Ambassador Duck Club and Public Shooting Grounds sediments.

SAV leaf tissue nutrient concentrations collected during the summer and fall months showed consistent levels of organic carbon (C) between nutrient enriched and reference ponds (37.8–41.7% C, Figures 3 and 4). Concentrations of total nitrogen (N) in tissue were generally higher in nutrient enriched ponds (3.3–4.8% N at Ambassador, ISSR and Newstate), while FB WMA and PSG nitrogen levels ranged from 1.9–2.9% N during the summer. During the fall, SAV tissue nitrogen levels ranged from 3.6–4.1 at Ambassador

and FB WMA and 3.0–3.2 at the PSG references sites. Total phosphorus (P) levels of SAV leaves ranged from 0.41–0.70% P at nutrient enriched ponds during the summer and fall (except FB WMA T1 levels were 0.22 during the fall) whereas they ranged between 0.20–0.25 at the PSG references sites during both sampling periods.

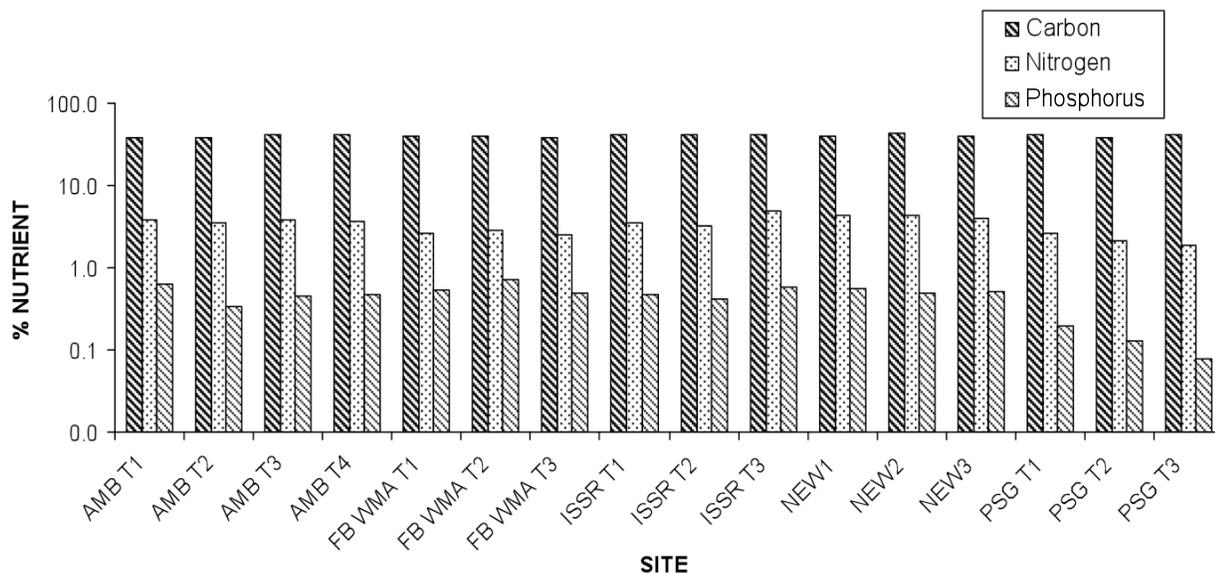


**Figure 2**—Mean sediment phosphorus (P) concentrations at the nutrient enriched and reference (PSG) impoundment sites. T1–T4 represent transect in sequential order of flow through the ponds; AMB = Ambassador, WMA = FB WMA, News = Newstate.

When C:N:P ratios of similarly-aged SAV leaves are compared between nutrient enriched and reference sites during July, all but one nutrient enriched site (Ambassador T2, pond 100) show carbon limitation and all sites (both nutrient enriched and reference) show nitrogen limiting ratios according to Redfield C:N:P ratios of organic matter,

106:16:1 (Table 2; Redfield 1934). By late fall, most nutrient enriched sites lacked enough plants to provide enough leaf sample for analysis or were lacking plants altogether. Those that had plants, showed even lower carbon ratios (with the exception of a gain in both carbon and nitrogen above limiting levels at FB WMA T1, Unit 1). The only samples that had SAV leaf C:N:P ratios that met or exceeded Redfield ratios were from summer PSG T2 and T3, fall FB WMA T1 and fall PSG T2.

The percent cover data indicate that SAV was impacted by nutrient loading primarily late in the season (Figure 5). The upstream or upper ponds of the impounded sites (with the exception of ISSR, which does not exhibit a flow-through series of ponds) are the first to receive flows from their respective source waters. Seasonal biological sampling revealed substantial differences in plant community responses in the upper ponds (Figure 5,  $F_{(df 4,8)} = 75.5, 13.6$ ;  $p$  value < 0.0001). The highest mean percent cover of SAV in the upper ponds at the nutrient enriched sites (i.e., those around Farmington Bay: Ambassador,  $75.2 \pm 8.2$ ; FB WMA,  $9.0 \pm 1.6$ ; Newstate,  $70.2 \pm 10.6$ ; and ISSR,  $89.9 \pm 2.4$ ) occurred in June and July but had declined substantially by August and was nearly absent by November ( $2.0 \pm 0.4, 2.4 \pm 0.4, 1.0 \pm 0.0, 5.8 \pm 1.1$ , respectively; Figure 5, mean  $\pm$  se). Reference SAV percent cover at the upper pond of PSG, on the other hand, remained quite high through November ( $95.8 \pm 0.6, 70.3 \pm 11.3, 93.7 \pm 2.7$  in July, August, and December, respectively).

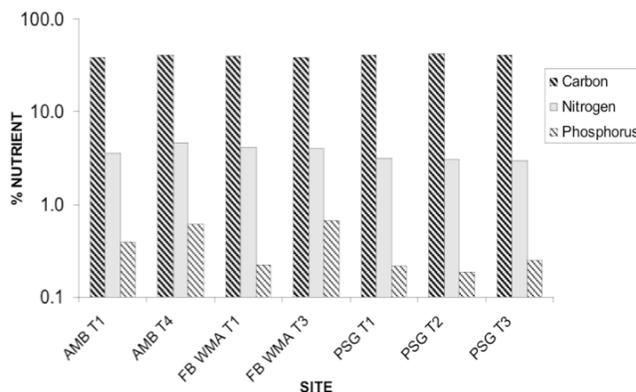


**Figure 3**—Percent carbon, nitrogen and phosphorus in submerged aquatic vegetation (SAV) during the summer, 2005. Numbers represent transect in sequential ponds from upstream to downstream; AMB = Ambassador Duck Club, NEW = Newstate Duck Club, PSG are reference ponds.  $n = 1$  at all nutrient enriched sites,  $n = 3$  at reference sites (PSG).

The multivariate factor analysis also supported the idea that the loss of SAV was related to nutrient enrichment. The principle axis determined by the analysis produced a water quality factor gradient showing nutrients and oxygen levels at one end, and high salinity (conductivity, TDS) at the other (Figure 6). Most of the impounded sites of this study showed moderate to abundant percent cover SAV in the early through late summer and there was no significant correlation with the principal multivariate gradient (data not shown;  $p$  value = 0.364). By the fall, however, most sites showed a decline in percent cover that corresponded with the water quality factor axis (Figure 6,  $F_{(df 1)}$ ,  $p$  value = 0.026).

## DISCUSSION

Seeds, tubers and vegetative parts of *Stuckenia* spp. are all preferred food by various waterfowl (Kantrud 1990) and indeed, many tens of thousands of waterfowl have been observed foraging and resting on these and other ponds each fall (Paul & Manning 2002). However, our data indicates that the *Stuckenia* density in many of the nutrient enriched ponds remained at low densities or experienced considerable senescence by the beginning of the fall waterfowl migration season, although there have been no apparent indicators of weakened birds (i.e. starvation, failure to migrate, massive downings, etc.). This suggests that either the birds are moving to more productive ponds on a regular basis or a significant amount of foraging on less preferred items or a combination thereof is occurring. Nonetheless, it remains important to elucidate waterfowl diets during this critical time on Great Salt Lake wetlands.



**Figure 4**—Percent carbon, nitrogen and phosphorus in submerged aquatic vegetation (SAV) during the fall, 2005. Numbers represent transect in sequential ponds from upstream to downstream; AMB = Ambassador Duck Club, NEW = Newstate Duck Club, PSG are reference ponds; (no plant material was available at Newstate Duck Club and the ISSR).  $n = 1$  at all sites except  $n = 3$  at PSG T1 and T3.

The apparent decline in vegetative productivity in the nutrient enriched ponds seem to contradict the paradigm that lower nutrient concentrations should result in less biomass and more nutrients should support greater biomass. There are two important factors that explain this contradiction:

1) Both emergent and submergent vegetation can derive all of their N and P requirements from sediments (Thiebaut & Muller 2003; Carr & Chambers 1998; Madsen & Adams 1988; Carignan & Kalff 1980). Indeed, Canfield & Hoyer 1988 and Peltier & Welch (1969) found no relationship between macrophyte growth and water-column P and N concentrations. Carignan & Kalff (1980) reported that nine common species of aquatic macrophytes, including *Stuckenia pectinata*, took all of their phosphorus from the sediments when grown in situ in both a mesotrophic and a mildly eutrophic bay. Even under hypereutrophic conditions, the sediments contributed an average of 72% of all the phosphorus taken up during growth. Therefore, submerged macrophytes in PSG obtain adequate nutrients from their associated sediments regardless of nutrient levels in the water column.

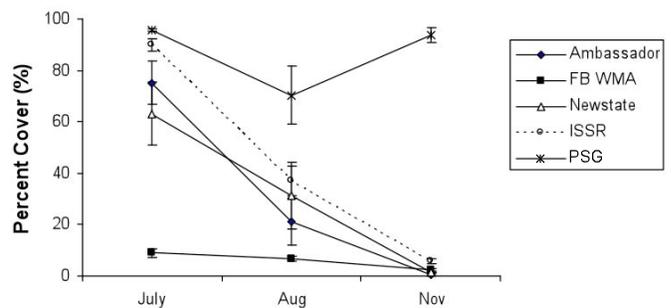
2) There is considerable evidence that the early senescence and loss of percent cover in the nutrient enriched ponds and particularly in comparison to the reference ponds at the PSG are the result of degraded water quality and related effects rather than normal seasonal changes. Total water column P in the nutrient enriched impounded sites was frequently more than an order of magnitude greater than in the PSG ponds and the resulting algal blooms may be the driving factor that is overwhelming those systems. Often heavy epiphytic biofilms (including sediment) were observed on the leaves of the SAV, and floating and entangled mats of macroalgae (Chlorophyta and Cyanophyta) and duckweed were frequently present at the nutrient enriched sites where nutrients were elevated. The “premature” senescence of SAV was likely induced by shade-related stress to the SAV by the epiphytic and macroalgal communities, and duckweed in some cases. Additionally, as percent cover of the SAV declines, suspended sediment from the wind events or carp perturbations remains in the water column for longer periods since there is no physical structure (i.e. plants) to slow water currents and facilitate settling and water clarity (Short & Short 1984; Ward et al. 1984). This turbidity may cause additional stress on the remaining SAV due to reduced light.

**Table 2**—C:N:P of Summer and Fall SAV Leaf Tissue for Reference and Nutrient enriched Sites, 2005; n = 3 for all sites. (Ratios with ☒ in front of them represent samples that met or exceeded the Redfield C:N:P ratios of 106:16:1, while ratios with !! in front if they approached Redfield ratios with respect to nitrogen levels. Asterisks identify sites that were dominated by *Ruppia cirrhosa*; all others were dominated by *Stuckenia* sp. Samples were taken in December, - indicates not enough above ground tissue for sampling.)

SITE	SUMMER (July)	FALL (November)
Ambassador_T1	61:6:1	-
Ambassador_T2	111:10:1	97:9:1
Ambassador_T3	89:8:1	-
Ambassador_T4*	87:8:1	67:7:1
FB WMA_T1	75:5:1	☒ 176:19:1
FB WMA_T2	55:4:1	-
FB WMA_T3	79:5:1	58:6:1
ISSR_T1	86:7:1	-
ISSR_T2*	100:8:1	-
ISSR_T3*	73:8:1	-
Newstate_T1	72:8:1	-
Newstate_T2	90:9:1	-
Newstate_T3	76:8:1	-
PSG_T1	!! 208:14:1	!! 188:14:1
PSG_T2	☒ 291:16:1	☒ 220:16:1^
PSG_T3	☒ 540:24:1	161:12:1

Although somewhat lower, sediment P concentrations in PSG were in the same range as those in the nutrient enriched ponds (approximately 300–500 mg kg<sup>-1</sup>) yet SAV did not show a premature senescence as in the reference sites. Madden & Kemp (1996) simulated eutrophication in submersed estuarine plant communities. They noted several important responses under nutrient enriched conditions that may have implications for the Farmington Bay nutrient enriched sites. In their study, epiphytic algal biomass was stimulated by an order of magnitude, while SAV biomass declined severely under both N + P enrichment. Phosphorus enrichment alone has not been shown to trigger community shifts in estuarine production but when N + P enrichments are introduced to mesocosm experiments and model simulations, epiphyte production can be exponential, while attenuating light to deleterious levels to SAV (Madden & Kemp 1996; Taylor et al. 1995, 1999). Dissolved inorganic nitrogen (NO<sub>2</sub>- and NO<sub>3</sub>-) levels in two of the nutrient enriched ponds (Ambassador and Newstate) were high, whereas those in the other two nutrient enriched wetlands and in the reference wetland were low, and likely limiting for algal growth. However, cyanobacteria provide enough

fixed nitrogen locally to support heavy epiphytic growth (Powell et al. 1989). Additionally, N<sub>2</sub>-fixing heterotrophs and cyanobacteria associated with duckweed mats have been found to fix as much as 15-20% of the nitrogen requirement for duckweed (Zuberer 1982), a substantial amount that could also contribute to the localized water column pool for macroalgae and diatoms. The increased density and coverage in duckweed, and filamentous and epiphytic algae in response to increased nutrients has been well documented (Vaithyanathan & Richardson 1999; Portielje & Roijackers 1995, and others). Accordingly, where nitrogen is limited, rich populations of epiphytic, nitrogen-fixing cyanobacteria most often accompany duckweed populations (Duong & Tiedje 1985, Zuberer 1982; Finke & Seeley 1978). This ability to manipulate nutrient availability provides a symbiotic relationship that favors a floating duckweed community. In turn, increased shading and concomitant increased tendency toward anoxia deeper in the water column may severely restrict health and survival of submerged vegetation (Morris et al. 2004). Further, Madden & Kemp (1996) found that in nutrient enriched treatments epiphytic growth on SAV increased and that the leaf tissue area decreased due to leaf mortality and sloughing. The periphytic growth was an important factor in the decline of SAV due to increased shading—more so than turbidity related to phytoplankton blooms.



**Figure 5**—Mean percent areal cover of SAV during the summer and fall months of 2005 for both nutrient enriched (Ambassador, FB WMA, Newstate, and ISSR) and reference (PSG) upper ponds (n = 10; ± se; p value < 0.0001).

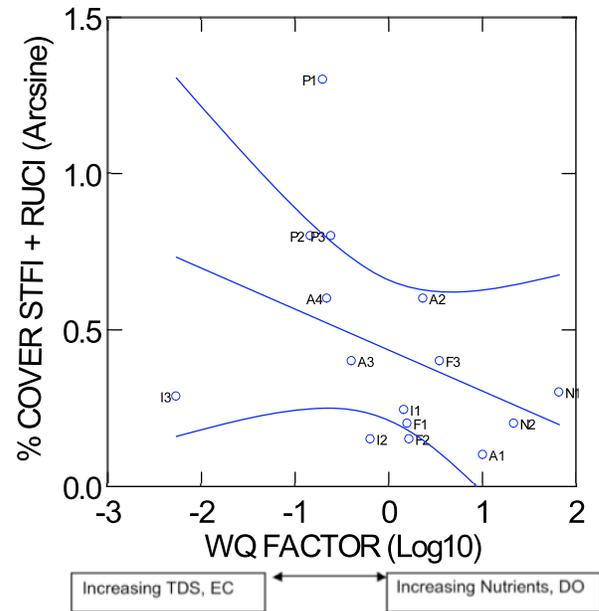
Another important conclusion from Madden & Kemp (1996) was that long-term shading stress to SAV in enriched environments inhibits carbon storage in root and rhizome tissues. SAV roots and rhizomes can provide a root buffering effect such that carbon stored from production periods is reserved for reproduction the following spring. When Madden & Kemp (1996) ran their model for successive years under sustained nutrient enrichment, detrimental epiphyte loads lead to negative P:R (production to respiration ratio) and resulted in reduced SAV biomass, reduced carbon stored in the roots and rhizomes, and

ultimately a decreased reproductive potential. They concluded that a “root buffering effect” is essential for long-term survival of SAV beds and to restore plants to historic levels would likely require improvements to water quality that persist for several years to allow root rhizome systems to become re-established.

The similarity of sediment P concentrations between nutrient enriched and reference sites, and yet vastly different water column P concentrations between nutrient enriched and reference sites, provides evidence for the connection between nutrient enrichment and various indicators of severe eutrophication in Farmington Bay nutrient enriched ponds. In turn, it is likely that epiphytic growth on the SAV leaves and presence of algal mats and duckweed attenuated light below critical levels required by *Stuckenia* and lead to a premature senescence of SAV. This condition was likely exacerbated as fall progressed and photoperiod and sun angle diminished. During fall collections at nutrient enriched ponds, SAV roots and rhizomes of remaining shoots were often rotting or not well developed (the only exceptions were ISSR T2 (West Pond A) and T3 (Southwest Pond South) and Ambassador T3 (W2) and T4 (W5) where *Ruppia cirrhosa* dominated; Hoven, personal observation). With reduced photosynthetic capacity and resultant reduction in oxygen transport to the roots, below ground tissue may have been susceptible to sulfide toxicity and/or infection by pathogens such as slime mold. On the other hand, plants at the PSG reference ponds grew densely and were difficult to pull (i.e. their roots and rhizomes were well developed and strong) as late as December. Although the plants are perennial, it is possible that roots and rhizomes of SAV at nutrient enriched sites lack carbon stores to regenerate each spring and rely heavily on seedlings each year to maintain the beds.

During the summer, N and P concentrations in SAV leaves reflect differences in available N and P levels in the water (Figures 3 and 4, Tables 1). In particular, SAV assimilated high levels of P at the nutrient enriched sites where P is elevated in the water and sediment, and maintained low levels of P at the reference sites both during the summer and fall. It is possible that higher levels of P in the nutrient enriched SAV samples reflected incomplete removal of periphyton from those leaves, however the majority of the periphyton community was composed of loosely attached diatoms (Rushforth & Rushforth 2007) and likely removed during sample processing. While C:N:P ratios appear to indicate nutrient imbalances in the SAV among ponds, it is difficult to identify the driving nutrient without additional information (Table 2). For example, what may appear to be carbon limitation in nutrient enriched ponds (i.e. low C:P ratios) may simply be that high P levels in the SAV are overwhelming the C levels. Alternatively, apparent high

C:P ratios in the tissue may be indicative of P starvation. For this reason, C:N:P ratios alone are not a suitable metric of wetland health but may serve as a supplementary metric. Presumably, the improved tissue nutrient (C:N:P) ratios at FB WMA Unit 1 during the fall may reflect nutrient inputs from the pond’s use as a rest area for waterfowl. Waterfowl grazing on SAV in this pond may have stimulated new shoots that could have had lower epiphyte burdens in the fall and better photosynthetic capacity to fix carbon than during the summer months. However, there was limited SAV cover at that time of year (Figure 6).



**Figure 6**—Percent cover ( $\pm$  95% confidence interval) of STFI + RUCI (SAV) versus WQ factor (see text) at nutrient enriched and reference ponds during November 2005 ( $p$  value = 0.026). A = Ambassador Duck Club, F = Farmington Bay Wildlife Management Area, I = Inland Sea Shorebird Reserve, N = Newstate Duck Club, P = Public Shooting Grounds (reference site). DO = dissolved oxygen. Numerals show the successive pond at each study area.

Our study shows that SAV percent cover in nutrient enriched ponds responded differently than that of reference ponds and that the nutrient enriched response over time is occurring at different rates relative to nutrient levels in the water (Table 1, Figures 5, 6). SAV, specifically *Stuckenia* sp., demonstrated a premature senescence during August, which amounted to 62–84% loss in areal cover at nutrient enriched ponds. Notably, this was before the arrival of waterfowl migrants. The largest numbers of ducks were accounted for in managed wildlife areas of Great Salt Lake during September as they migrate southward for winter the months (Paul & Manning 2002). Extensive surface mats of filamentous algae or duckweed often developed on these ponds and heavy coatings of biofilms (composed of epiphytic algae, sediment, and possibly bacteria and fungi) were observed on the living leaves. This surface epiphytic

and biofilm shading may reduce light penetration to below optimal or even threshold requirements (Taylor et al. 1995, 1999; Madden & Kemp 1996). Further, this would expectedly be exacerbated by shorter photoperiod and lower sun angle as fall progressed. If photosynthesis rates are sub-optimal (i.e.  $P < R$ ), there may not be adequate oxygen production to diffuse down to the roots and maintain an oxygen-rich root zone to sustain healthy plants. In turn, this could represent a decline in additional food availability during a time when waterfowl are attempting to nourish and regain energy stores.

Early senescence and its cause(s) warrant further investigation, particularly as to the seasonal timing and whether correlations between nutrient (water column and SAV tissues), light attenuation, and biomass or extensive biofilms exist. If such correlations are documented, they need to be quantified and considered for inclusion into an index of biotic integrity. In addition, the literature indicates that Photosystem II fluorescence is a useful indicator of plant stress (in this case, stress from shading), and exhibits potential as an SAV community metric of wetland condition that may help explain the premature senescence of the SAV (Baker 2008). Together, these metrics may serve to evaluate the condition of impounded wetlands relative to water quality. Ultimately, with regard to the beneficial uses, our concern is that heavy nutrient loading to Farmington Bay's wetlands are causing the disappearance of large underwater meadows of *Stuckenia* (a preferred food by waterfowl) prior to the arrival of migrating of waterfowl.

## REFERENCES

- Anscombe, F.J. 1948. The transformation of Poisson, binomial, and negative binomial data. *Biometrika* 35: 246–254.
- ASTM D5373. 2002. Standard test methods for instrumental determination of carbon, hydrogen, and nitrogen in laboratory samples of coal and coke.
- Baker, N.R. 2008. Chlorophyll fluorescence: a probe of photosynthesis in vivo. *Annual Reviews of Plant Biology* 59: 89–113.
- Brix, H. & J.E. Lyngby. 1983. The distribution of some metallic elements in eelgrass (*Zostera marina* L.) and sediment in the Limfjord, Denmark. *Estuarine and Coastal Shelf Science* 16: 455–467.
- Burrell, D.C. & J.R. Schubel. 1977. Seagrass ecosystem oceanography. In: McRoy, C.P. & C. Helffferich (eds), *Seagrass ecosystems: a scientific perspective*. Marcel Dekker, New York: 195–232.
- Buzzelli, C.P., R.L. Wetzel and M.B. Meyers. 1999. Dynamic simulation of littoral zone habitats in lower Chesapeake Bay. II. Seagrass habitat primary production and water quality relationships. *Estuaries* 21: 673–689.
- Canfield, D.E. & M.V. Hoyer. 1988. Influence of nutrient enrichment and light availability on the abundance of aquatic macrophytes in Florida streams. *Canadian Journal of Fisheries and Aquatic Science* 45: 1467–1472.
- Carignan, R. & J. Kalff. 1980. Phosphorus sources for aquatic weeds: Water or sediments? *Science* 207: 987–989.
- Carr, G.M. & P.A. Chambers. 1998. Macrophyte growth and sediment phosphorus and nitrogen in a Canadian prairie river. *Freshwater Biology* 39: 525–536.
- Daubenmire, R.F. 1959. A canopy-coverage method of vegetational analysis. *Northwest Science* 33(1): 43–61.
- Duong, T.P. & J.M. Tiedje. 1985. Nitrogen fixation by naturally occurring duckweed-cyanobacterial associations. *Canadian Journal of Microbiology* 31: 327–330.
- EPA 325.2. 1978. Methods for the chemical analysis of water and wastes (MCAWW) (EPA/600/4-79/020).
- EPA 353.2. 1993. Methods for determination of inorganic substances in environmental samples (EPA/600/R-93/100).
- EPA 365.1. 1993. Methods for determination of inorganic substances in environmental samples (EPA/600/R-93/100).
- EPA 365.2. 1993. Methods for determination of inorganic substances in environmental samples (EPA/600/R-93/100).
- Finke, L.R. & H.W. Seeley. 1978. Nitrogen fixation (acetylene reduction) by epiphytes of freshwater macrophytes. *Applied and Environmental Microbiology* 36: 129–138.
- Hall, M.O., M.J. Durako, J.W. Fourqurean & J.C. Zieman. 1999. Decadal changes in seagrass distribution and abundance in Florida Bay. *Estuaries* 22(2B): 445–459.
- Hoven, H.M., H.E. Gaudette & F. T. Short. 1999. Isotope ratios of  $^{206}\text{Pb}/^{207}\text{Pb}$  in eelgrass, *Zostera marina*, indicate sources of Pb in an estuary. *Marine Environmental Resources* 48: 377–387.
- Kantrud, H. 1990. Sago pondweed (*Potamogeton pectinatus* L.) a literature review. U.S. Fish and Wildlife Service, Northern Prairie Wildlife Research Center, Jamestown North Dakota, USA.
- Kemp, W.M., R.R. Twilley, W.R. Boynton & J.C. Means. 1983. The decline of submersed vascular plants in upper Chesapeake Bay: Summary of results concerning possible causes. *Marine Technology Society Journal* 17: 78–79.
- Kenworthy, W.J., J.C. Zieman & G.W. Thayer. 1982. Evidence for the influence of seagrasses on the benthic nitrogen cycle in a coastal plain estuary near Beaufort, North Carolina (USA). *Oecologia* 54: 152–158.
- Madden, C.J. & W.M. Kemp. 1996. Ecosystem model of an estuarine submersed plant community: calibration and simulation of eutrophication responses. *Estuaries* 19: 457–474.
- Madon, S.P. 2006. Analyses of 2005 data on wetland biota and water quality in Farmington Bay, Great Salt Lake, Utah. Final technical memorandum 2, CH2MHill Inc., Salt Lake City, Utah.
- Madsen J.D. & M.S. Adams. 1988. The nutrient dynamics of a submersed macrophyte community in a stream ecosystem dominated by *Potamogeton pectinatus*. *Journal of Freshwater Ecology* 4: 541–550.

- Miller, T.G. & H.M. Hoven. 2007. Ecological and beneficial use assessment of Farmington Bay Wetlands: Assessment and site-specific nutrient criteria methods development Phase I. Progress Report to EPA, Region VIII and Final Report for Grant: CD988706-03, 760 pp.
- Morris, K., K.A. Harrison, P.C.E. Bailey & P. Boon. 2004. Domain shifts in the aquatic vegetation of shallow urban lakes: the relative roles of low light and anoxia in the catastrophic loss of the submerged angiosperm *Vallisneria americana*. *Marine and Freshwater Research* 55: 749–758.
- Olson, B.E., K. Lindsey & V. Hirschboeck. 2004. Habitat management plan: Bear River Migratory Bird Refuge, Brigham City Utah. U.S. Department of the Interior Fish and Wildlife Service, Brigham City, Utah, 191 pp.
- Orthe, R.J. & K.A. Moore. 1983. Chesapeake Bay: An unprecedented decline in submerged aquatic vegetation. *Science* 222: 51–53.
- Paul, D.S. & A.E. Manning. 2002. Great Salt Lake waterbird survey five-year report (1997–2001). Utah Division of Wildlife Resources, Salt Lake City.
- Peltier, W.H. & E.B. Welch. 1969. Factors affecting growth of rooted aquatics in a river. *Weed Science* 17: 412–416.
- Phillips, R.C. & E.G. Meñez. 1988. Seagrasses. *Smithsonian Contributions to the Marine Sciences # 34*, 104 pp.
- Portielje R & R.M.M. Roijackers. 1995. Primary succession of aquatic macrophytes in experimental ditches in relation to nutrient input. *Aquatic Botany* 50:127–140.
- Powell, G.V.N., W.J. Kenworthy & J.W. Fourqurean. 1989. Experimental evidence for nutrient limitation of seagrass growth in a tropical estuary with restricted circulation. *Bulletin of Marine Science* 44: 324–340.
- Redfield, A.C. 1934. On the proportions of organic derivations in seawater and their relation to the composition of plankton. In: Daniel, R.J. (ed), James Johnson Memorial Volume. University Press of Liverpool, pp. 177–192.
- Rushforth, S.R. & S.J. Rushforth. 2007. A taxonomic and bioassessment survey of the diatom floras of Farmington Bay, Great Salt Lake 2005; Rushforth Phycology, LLC, Orem, Utah, 122 pp.
- Short, F.T. & D.M. Burdick. 1996. Quantifying eelgrass habitat loss in relation to housing development and nitrogen loadings in Waquoit Bay, Massachusetts. *Estuaries* 19: 730–739.
- Staver, L.W., K.W. Staver & J.C. Stevenson. 1996. Nutrient inputs to the Choptank River estuary: implications for watershed management. *Estuaries* 19: 342–358.
- Stumpf, R.P., M.L. Frayer, M.J. Duako & J.C. Brock. 1999. Variations in water clarity and bottom albedo in Florida Bay from 1985 to 1997. *Estuaries* 22: 431–444.
- Taylor, D.I., S.W. Nixon, S.L. Granger & B.A. Buckley. 1995. Nutrient limitations and the eutrophication of coastal lagoons. *Marine Ecology Progress Series* 127: 235–244.
- Taylor, D.I., S.W. Nixon, S.L. Granger, & B.A. Buckley. 1999. Responses of coastal lagoon plant communities to levels of nutrient enrichment: a mesocosm study. *Estuaries* 22: 1041–1056.
- Thayer, G.W., S.M. Adams & M.W. Lacroix. 1975. Structural and functional aspects of a recently established *Zostera marina* community. In: *Estuarine Research*, L.E. Cronin, (ed). Academic Press, New York 1: 518–540.
- Thayer, G.W., W.J. Kenworthy & M.S. Fonseca. 1984. The ecology of eelgrass meadows of the Atlantic Coast: a community profile. U.S. Fish and Wildlife Service, /OBSO-84/02.
- Thiebaut, G. & S. Muller. 2003. Linking phosphorus pools of water sediment and macrophytes in running waters. *Annales de Limnologie–International Journal of Limnology* 39: 307–316.
- Tomasko, D.A., C.J. Dawes & M.O. Hall. 1996. The effects of anthropogenic nutrient enrichment on turtle grass (*Thalassia testudinum*) in Sarasota Bay, Florida. *Estuaries* 19: 448–456.
- Vaithyanathan, P. & C.J. Richardson. 1999. Macrophyte species changes in the Everglades. Examination along a eutrophication gradient. *Journal of Environmental Quality* 28: 1347–1358.
- Vaiela, I., J. McClelland, J. Hauxwell, P.J. Behr, D. Hersh & K. Foreman. 1997. Macroalgal blooms in shallow estuaries; controls and ecophysiological and ecological consequences. *Limnology and Oceanography* 42: 1105–1118.
- Ward, T.J. 1987. Temporal variation of metals in the sea grass *Posidonia australis* and its potential as a sentinel accumulator near a lead smelter. *Marine Biology* 95: 315–321.
- Wolfe, D.A., G.W. Thayer & S.M. Adams. 1976. Manganese, iron, copper and zinc in an eelgrass (*Zostera marina* L.) community. *Radiology and Energy Resources. Proceedings of the Fourth Ecological Society of America, Special Publication 1*: 256–270.
- Zieman, J.C., J.W. Fourqurean & T.A. Frankovich. 1999. Seagrass dieoff in Florida Bay: Long-term trends in abundance and growth of turtle grass, *Thalassia testudinum*. *Estuaries* 22(2B): 460–470.
- Zuberer, D.A. 1882. Nitrogen fixation (acetylene reduction) associated with duckweed (Lemnaceae) mats. *Applied and Environmental Microbiology* 43: 823–828.